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Urbanization as a land use change driver of forest ecosystem services S. Delphin^{a,1}, F.J. Escobedo^{b,*}, A. Abd-Elrahman^c, W.P. Cropper^d

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abstract
Land use change in the form of urbanization is a direct driver affecting the provision of ecosystem services from forests. To better understand this driver, we modeled the effects of urbanization on three regulating and provisioning ecosystem services in two disparate watersheds in Florida, USA. The study integrated available geospatial and plot-level forest inventory data to assess future changes in carbon storage, timber volume and water yield during a period of 57 years. A 2003–2060 urbanization and land use change scenario was developed using land cover data and a population distribution model. The Integrated Valuation and Ecosystem Services Tradeoffs model was then used to quantify changes in ecosystem services. Carbon storage was reduced by 16% and 26% in the urbanized 2060 scenario in both the rural Lower Suwannee and urban Pensacola Bay watersheds, respectively. Timber volume was reduced by 11% in the Lower Suwannee and 21% in the Pensacola Bay watershed. Water yield, however, increased in both watersheds by 4%. Specific sub-watersheds that were most susceptible to urbanization were identified and mapped and ecosystem service interactions, or trade-offs and synergies, are discussed. Findings reveal how urbanization drives the spatio-temporal dynamics of ecosystem services and their trade-offs. This study provides policy makers and planners an approach to better develop integrated modeling scenarios as well as designing mapping and monitoring protocols for land use change and ecosystem service assessments. © 2016 Elsevier Ltd. All rights reserved.

1. Introduction The world's population is rapidly increasing (US Census Bureau, 2013) and the world's population is projected to be 9 billion by 2050. This population increase will lead to land use and cover (LULC) changes and alter the provision of ecosystem services such as food, timber, and clean water (European Commission, 2009; Hoyer and Chang, 2014; Millennium Ecosystem Assessment, 2003; Vitousek et al., 1997). Urbanization, the development of new urban areas from non-urban lands, is a key anthropogenic driver affecting ecosystems (Bengston et al., 2005; He et al., 2016; Yue et al., 2013; Zhang et al., 2012). The use of the driver concept in influenc-

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ing land use change and ecosystem services, has become a common approach for studies analyzing policies and management objectives and associated effects on human well-being (Hoyer and Chang, 2014; Yue et al., 2013). By analyzing ecosystem service drivers using

integrated modeling scenarios, as well as measured and available data, we can better understand ecosystem function and changes in their services. We can also provide information for planning and policy formulation in order to minimize potential negative impacts to human well-being (Millennium Ecosystem Assessment, 2003). Forests are important for carbon storage and this function is referred to as a regulating ecosystem service as it regulates global climate (Millennium Ecosystem Assessment, 2003). Forest carbon storage is commonly measured according to four pools: aboveground biomass, belowground biomass, litter or dead biomass, and soil (Ashton et al., 2012). Indeed, forests are critical to the global carbon cycle as they comprise a large stock of carbon relative to other ecosystems; thus conserving peri-urban forested areas from urbanization is a priority (Gorte, 2009; Harmon, 2001; Jandl et al., 2007; Yonavjak et al., 2011). Urbanization of forests in the southern U.S. has for example resulted in about 0.21 Petagrams (Pg) of carbon emissions from 1945 to 2007 (Zhang et al., 2012). Zhang and

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Nagubadi (2005) also found that urbanization is one of the principal drivers in the decline of natural forest timber areas. Urban areas can store carbon in the form of planted and remnant trees; but, 70–100 years are needed before pre-urbanization carbon stocks are reached (Zhang et al., 2012). Furthermore, the recovery of carbon stocks to pre-urbanization levels is uncertain and depends on the specific ecosystem, its structure and management of the existing urban forest. As such, timber supplies are significantly affected by urbanization. Timber products are considered a provisioning ecosystem service that provides direct benefits to people (Boyd and Banzhaf, 2007; Millennium Ecosystem Assessment, 2003). The benefits of these provisioning ecosystem services (e.g., timber, food, nontimber forest products, water supply) are widely recognized and are more easily valued than most other ecosystem services because of the existence of markets (De Groot et al., 2010). For example, forests in the Southern U.S. produce more than 60% of the nation's timber products and substantially contribute to the region's economy (Prestemon and Abt, 2002a). Bystriakova et al. (2005) report that timber harvesting at the global scale has increased by 60% in the last four decades and the increasing trend is expected to continue, but at a slower rate. Further, Cubbage et al. (1995) found that urbanization reduced timber supply, especially of softwoods and similarly Hodges et al. (1998) has documented how forest-timber areas in Louisiana, U.S. have been reduced by urban development. Water is another valuable forest regulating and provisioning ecosystem service (Kreye et al., 2014). Land use changes can however alter the quantity and quality of water (Foley et al., 2005; Sun et al., 2005); therefore, determining the effect of forest LULC changes on water yield and quality is an important consideration. Water yield is the sum of annual precipitation that does not evaporate from soil and water or transpire from vegetation (Mendoza et al., 2011). Since urban areas increase runoff, as a result of decreased interception, evapotranspiration and infiltration, they can decrease water quality and increase overall water yield (Arnold et al., 1987; Hanson et al., 2010). Indeed several studies have determined the relationship between the hydrological cycle, forests (Bosch and Hewlett, 1982; Sun et al., 2005), and urbanization. Hollis (1977) reports that median flow increased by approximately 40% post-urbanization in the Canon's Brook catchment in England. Arnold et al. (1987) also modeled the effect of urbanization on water yield in a rural Texas U.S. watershed that urbanized over 70 years to about 77% urban land cover and found that annual surface runoff would increase by about 10%. Other studies in Florida, U.S. have analyzed the effect of urbanization on water quality (Cooley and Martin, 1979; Frick, 1998) and water supply (Boggess, 1968). However, we found no other

relevant studies on the effects of urbanization on forest water yield or timber production in the State of Florida. Geographic Information Systems (GIS) models have been used to assess the potential effects of different drivers on the provision of ecosystem services in subtropical forests (Cademus et al., 2014; Metzger et al., 2006). Timilsina et al. (2011), for example, used geospatial models to map forest carbon storage hotspots in Florida U.S. and identified plot-level ecological and anthropogenic drivers of this regulating ecosystem service, based on the Millennium Ecosystem Assessment (2003) classification. Cademus et al. (2014) also used geospatial and forest inventory data to assess interaction or the trade-offs (win-lose situations), synergies (win-win situations) and drivers in pine dominated forests in Florida. Models such as the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) developed by the Natural Capital Project (<http://www.naturalcapitalproject.org/>), have also used available geospatial data to estimate and map the provision of ecosystem services (Tallis et al., 2011). The model estimates the provision of ecosystem services in a current land use/cover and in different scenarios that incorporate proposed policies, management goals and disturbance regimes. Because of this, the InVEST model has been recently used to quantify and value ecosystem services and thus facilitate their integration into policy making (Daily et al., 2009; Hoyer and Chang, 2014; Tallis et al., 2011). Daily et al. (2009) and Nelson et al. (2009) have applied InVEST in different areas of the US, and He et al. (2016) in China, to assess the effects of alternative land use and management decisions on ecosystem services. Hoyer and Chang (2014) estimated the provision of freshwater ecosystem services for two basins in Oregon using several urbanization and climate change scenarios. Delphin et al. (2013) and Bottalico et al. (2016) also analyzed the effect of a driver (i.e., hurricanes) and forest management scenarios, respectively, on two ecosystem services, timber and carbon stocks, using the InVEST and other geospatial models. Approaches such as those of Delphin et al. (2013), Bottalico et al. (2016), He et al. (2016); and Hoyer and Chang's (2014) could therefore be adapted to assess both the spatial and temporal effects of urbanization on multiple ecosystem services and their interactions as well as to better formulate land use policies. Given the lack of studies on the effects of ecosystem service drivers on subtropical forest ecosystem service provision, the aim of our study was to assess the effects of urbanization on a bundle of forest ecosystem services and their trade-offs: carbon storage as an indicator of climate regulation, water yield as an indicator of flooding regulation, and timber volume, as an indicator of a provisioning service. Our specific objectives were to quantify both the temporal and spatial effects of urbanization on these key ecosystem services from 2003 to 2060 using: (1) the InVEST model, (2) plot-level United States Department of Agriculture (USDA) Forest Inventory and Analysis (FIA) data, and (3) a regional forecasted urbanization land cover change scenario for 2003–2060 based on a population distribution spatial model. We used the rural Lower Suwannee River and urbanized Pensacola Bay watersheds in northwestern Florida U.S. as our two study areas. These two representative, yet disparate in terms of rural versus urban land uses, study watersheds were selected based on: the proportion of forest and urban land covers, FIA plot density, and the presence of impaired water bodies (i.e., do not meet water quality standards) according to Florida's Statewide Comprehensive Verified List of Impaired waters (<http://www.dep.state.fl.us/water/watersheds/assessment/a-lists.htm>). This integrated modeling approach using two subtropical watersheds should contribute to policy-relevant land management and planning assessments. It can also be used to better communicate the temporal and spatial effects and trade-offs of urban development and other land use changes on ecosystem services in other types of forested ecosystems in the world. 2.

Material and methods

2.1. Study areas

The rural Lower Suwannee River watershed in western Florida has 45% forest, 6% urban, and 25% agricultural land covers while the urbanized Pensacola Bay watershed in northwestern Florida has approximately 20% urban, 39% forest, and 0.1% agriculture land covers (Fig. 1). The Lower Suwannee watershed consists of 63 subwatersheds in an area of 408,828 ha. and is one of five watersheds that comprise the Suwannee River watershed (Katz et al., 1997). The Suwannee river is the second largest river in Florida in terms of average discharge (Light et al., 2002). The Pensacola Bay watershed comprises 15 sub-watersheds in an area of 140,825 ha. and is part of the Pensacola Bay system that is composed by five watersheds (Schwenning et al., 2007). As previously explained, the two 190 S. Delphin et al. / Land Use Policy 54 (2016) 188–199 Fig. 1. The Lower Suwannee River and Pensacola Bay watershed study areas in Florida U.S. watersheds were selected as they have been prioritized by the State of Florida for water quality protection.

2.2. Geospatial and field data

2.2.1. Florida vegetation and land cover 2003

We used the 43 vegetation and land cover classes from the 2003 Florida Vegetation and Land cover (FVLC) map developed by the Florida Fish and Wildlife Conservation Commission (FWC) based on Landsat Enhanced Thematic Mapper satellite imagery with a 30 meter spatial resolution (Stys et al., 2004). Carbon stores and timber volume were estimated for the 11 forest cover classes identified by FVLC data. Given that the scope of this analysis is forest land cover changes, forests or trees and other associated vegetation within already urbanized land covers, were not spatially or statistically analyzed. However, existing studies were used to explore carbon and water yield dynamics within urban forests and land covers. At the time of this study the 2003 FVLC map was the most detailed classification for the state of Florida.

2.2.2. 2060-Land use and land cover map

The forecasted urbanization 2060-LULC map was developed from the 2003-Florida Vegetation and Land Use map and a 2060- Florida Population Distribution model (Zwick and Carr, 2006). This model is based on population projections from the Bureau of Economic and Business Research (BEBR) and a GIS land use suitability analysis (Zwick and Carr, 2006). The data and models account for periods of increased economic activity and rapid urbanization (i.e., 2003–2008), economic downturn and decline in urbanization rate (i.e., 2008–2013), and planning projections beyond 2040 that are regularly used for urban and forest policy formulation in Florida (Wear and Greis, 2002). Three main criteria considered in the land use model were: (1) suitability to support a specific land use category, (2) preference to reflect community values, and (3) potential conflict with conservation (i.e., forests), urban and agricultural lands. Based on BEBR population projections, the model calculates the human population and land area that need to be allocated; it then identifies and maps suitable land use polygons for these new urban areas in Florida. Overall, polygons that are close to existing urban areas, higher road densities, and that lack wetlands and conservation lands were the ones most heavily weighted for land use change. Specific details, methods and assumptions can be found in Zwick and Carr (2006). The 2060-Population Distribution map contains the existing and projected urban areas, this result was then combined with the 2003-Florida Vegetation map to forecast and produce the 2060 urbanization scenario-land cover map.

2.2.3. Forest inventory and analysis program data

We used available USDA Forest Service Inventory and Analysis (FIA) program data to estimate forest carbon storage and timber volume as the 2003 current baseline for our urbanization scenario. The FIA provides tree and plot-level data for all forest lands in all the U.S including forest extent, characteristics, condition, and tenures (Jenkins et al., 2003; Woudenberg et al., 2010). We specifically used the FIA data from cycle 8 (2002–2007) since it accounts for our analysis period. To ensure

private landowners' privacy and plot integrity, geographic coordinates (latitude and longitude) of the plots are not provided because of a privacy provision enacted by the U.S. Congress in the Food Security Act of 1985. Therefore, plots are located within a buffer of approximately 0.8 to 1.6 km from their true centroid. Alternative approaches based on other FIA attributes, can however be applied to better approximate true plot centroids. For example, only the 1-10% of plots are misclassified when analyzing for ecological subsection data (Sabor et al., 2007). Thus, accounting for the ecological subsection strata, FIA survey unit and County attributes can better help specify true plot centroids. Accordingly, we used an approach described in Delphin et al. (2013) that uses specific FIA database attributes: FIA Survey Unit, ecological subsection, and county, to better approximate the true FIA plot centroid, according to unique combination of classes for each forest ecosystem, ecological subsection and county. Once forest ecosystems were identified for each watershed, the main forest tree species per watershed were identified based on vegetation descriptions prepared by Gilbert and Stys (2004) that contain the list of forest tree species observed in each forest ecosystem. Finally, the forest tree species found in the FIA database were considered and the species codes were identified.

2.2.4. Forest carbon storage estimates

The InVEST Carbon Storage and Sequestration model was used to analyze carbon storage in the 2003-baseline land covers. These carbon storage estimates were then projected to the 2060-urbanized land cover. The FIA tree and plot-level data described in Section 2.2.3 were used as input for the model, which estimates carbon storage in Megagrams (Mg) per hectare (ha) for each land cover. Carbon storage analyses were estimated for the different forest types in the 2003 Florida Vegetation and Land cover map. The four carbon pools (Mg per ha) were aboveground biomass, belowground biomass, soil organic matter and dead biomass (Appendix A; Timilsina et al., 2011). The FIA provided above and belowground carbon stores at the individual tree-level; therefore to obtain plot-level data, per-tree values were multiplied by an FIA-defined expansion factor to scale trees on a plot to their per acre equivalent (Woudenberg et al., 2010). Accordingly, carbon values per tree were summed by plot using species code (SPCD), FIA survey unit (UNITCD), ecological subsection (ECOSUBCD), county (COUNTYCD), condition status (COND STATUS CD), and stand age (STDAGE). Only plots with the condition status defined as forest land and with a STDAGE ranging from 1 to 72 years were considered in the analysis. We note that STDAGE is not the overall age of a stand but is an FIA indicator of the average live age of trees in a plot that are not overtopped in the predominant size-class for a given condition. Please see Appendix A for a description of carbon pools and Woudenberg et al. (2010) for a definition of FIA codes and variables. Timber harvest will reduce the amount of carbon stored in forests, however, the specific impact on landscape level carbon storage will depend on the type of silvicultural approach used (e.g., less frequent harvesting) (Schwenk et al., 2012). Because we were not able to account for this, no specific timber harvest activity effects were considered in the analysis. The FIA provides plot-level dead biomass and soil carbon estimates in tons per acre, which were then converted to megagrams per hectare. The carbon pools for the urbanized 2060 scenario were also estimated using an assumption that carbon pools in 2060 will be 5 percent less than 2003 pools (Huggett et al., 2013). There is significant uncertainty in future forest carbon projections, and so based on USDA Forest Service forecasts, this 5% reduction in annual forest growth for the southeastern region of the US was incorporated into our estimates (Heath and Smith, 2000; Huggett et al., 2013; Moore et al., 2002). This assumes that carbon stock estimates are a simple linear factor and thus they can easily be adjusted. The InVEST carbon storage model's intermediate output was the sum of carbon stocks in the four

pools: carbon aboveground (all living plant material above soil), belowground (living root systems below soil), dead (litter and standing dead biomass), and soil. Final model output was expressed in total Mg of carbon per pixel that were then converted to Mg per hectare. Data was classified using the ArcGIS Natural Breaks method (ESRI, 2011) to display high, moderate and medium range of change (2003–2060) for each sub-watershed.

2.2.5. Timber volume estimates Methods for estimating timber production were similar to those for carbon storage. Specifically, timber volume and timber in the sawlog portion were calculated as indicators of the potential timber available for harvesting. Plot-level timber volume was estimated using FIA survey unit, ecological subsection, and county. From the FIA tree-level data, two attributes were analyzed: the net volume (volnet) and the net volume in the saw-log portion (volsaw). The final results were converted to cubic meters per hectare and volume values were assigned to individual pixels in the map. For the 2060 timber volume projections, we used the volume values presented in Prestemon and Abt (2002b) and the Subregional Timber Supply Model (SRTS) of Abt et al. (2000), assuming an annual average increases of 1.7% to determine 2060 timber volume. Timber data were also classified based on the ArcGIS Natural Breaks method (ESRI, 2011) to identify high, moderate and medium ranges of change (2003–2060) for each sub-watershed.

2.2.6. Water yield estimates Water yield was estimated for a 10 year period using annual average precipitation (P) data for 1998–2008, from the PRISM Climate Group (PRISM Climate Group and Oregon State University, 2008). For 2060, P was estimated using the A2 scenario from Coulson et al.'s (2010) projections from the Australian Commonwealth Scientific and Industrial Research Organization's Mk3 Climate System. The A2 scenario is conservative for Florida as it represents lower socioeconomic and urban development rates, thus less conversion of forests to urban areas. Maximum soil and water table depth required by InVEST were kept constant for the analysis period and were obtained from the U.S. General Soil Map (State Soil Geographic-STATSGO2) developed by the USDA Natural Resources Conservation Service (USDA-NRCS) in 2006. Average annual Potential Evapotranspiration (PET) for the analysis period corresponded to 1998–2008 and was from the U.S. Geological Survey's Florida Integrated Science Center and plant available water content (AWC) data for the analysis period were from the USDA NRCS' General Soil Map-STATSGO2. Finally, maximum root depth was collected from Canadell et al. (1996) and the Zhang Constant (Z), a factor used to characterize the seasonality of precipitation in an area was from Tallis et al. (2011). Using this input data, the InVEST Water Purification model estimated the annual average water yield for a specific watershed. The model estimated water yield at the sub-watershed and watershed level and accounts for surface water and not ground water processes (Mendoza et al., 2011). Thus, there is no differentiation between surface, subsurface and base flow water in the InVEST model (Tallis et al., 2011). According to Zektser and Loaiciga (1993), groundwater comprises on average 30% of the stream flow in most hydrologic systems and in some areas can reach up to 90% (Winter et al., 1998). Therefore, in areas where groundwater is influential as in Florida US, the InVEST model can produce low water yield values when compared to observed stream flow levels (Mendoza et al., 2011).

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The InVEST water yield model is based on the Budyko curve and annual average precipitation (Tallis et al., 2011). The Budyko curve describes the relationship between precipitation and actual PET (Mendoza et al., 2011). This approach assumes that evapotranspiration depends on water availability (i.e., precipitation), atmospheric demand (i.e., PET; Mendoza et al., 2011; Zhang et al., 2001, 1999), and is used to explain the dependence of annual water balance on water and energy availabilities (Yang et al., 2011). The annual water yield is therefore estimated based on precipitation and actual

evapotranspiration. Specifically, the model analyzes each pixel considering each land cover according to Eq. (1) $Y_{xj} = j \times 1 - AET_{xj} P_{xj} \times P_{xj} \times A_{xj}$ (1) where Y_{xj} is the annual water yield, AET_{xj} is the annual actual evapotranspiration on pixel x with LULC $_j$, and P_{xj} is the annual precipitation in pixel x with LULC $_j$, and A_{xj} is the area of x in LULC $_j$, and accordingly AET , P and A were calculated for each LULC type. The evapotranspiration partition of the water balance is also calculated. This is an approximation of the Budyko curve according to Zhang et al. (2001) and is based on PET and plant available water content. The maximum difference in AET/P ratio corresponds to forest cover areas where treed areas have greater soil water storage (Zhang et al., 2001) and is described as: $AET_{xj} P_{xj} = 1 + w_{xj} R_{xj} \frac{1}{1 + w_{xj} R_{xj} + 1/R_{xj}}$ (2) where R_{xj} is the ratio of potential evapotranspiration to precipitation (i.e., Budyko dryness ratio) and w_{xj} is a modified dimensionless ratio of plant accessible water storage relative to expected precipitation during the year (Mendoza et al., 2011). w_{xj} is given by Eq. (3): $w_{xj} = Z AWC_x x_j$ (3) where AWC_x is the volumetric (mm) plant available water content, Z is the seasonality factor for rainfall distribution (Mendoza et al., 2011). Finally, the ratio of PET to precipitation (R_{xj}) is calculated by $R_{xj} = k_j \cdot ETo_x \frac{P_{xj}}{P_{xj}} = PET_{xj} P_{xj}$ (4) where ETo_x that is the reference evapotranspiration from pixel x and k_{xj} is the evapotranspiration coefficient associated with LULC $_j$ on pixel x . We note that Sánchez-Canales et al. (2012) observed that Z was less important than other inputs whereas plant evapotranspiration for each LULC was potentially a very sensitive parameter in the InVEST model. Water yield (2003–2060) was also classified using the ArcGIS Natural Breaks method (ESRI, 2011) to identify sub-watersheds with high, moderate and medium changes in water yield. The water yield model performance was also assessed by comparing average annual water yield from the model to average annual stream flow data from the Gopher River gage station (site 02323592) located near the Lower Suwannee watershed's outlet. Normally, 10 year average values of streamflow data should be used for water yield estimates. However, U.S. Geological Service (USGS) National Water Information System (NWIS) (<http://waterdata.usgs.gov/nwis/>), stream flow data were only available for 2000–2008 (Nielsen and Norris, 2007).

2.3. Statistical analysis

Statistical differences between the 2003 and the 2060 urbanization maps at the sub-watershed level were determined using a two tailed t-test with two paired samples for means at a 95% confidence level. Pearson correlation coefficients were calculated between carbon storage, timber volume, water yield, and the amount of forest areas. Negative correlation coefficients were also used to determine tradeoffs while positive coefficients determined synergies (Cadmus et al., 2014; Raudsepp-Hearne et al., 2010). All statistical analyses were at the sub-watershed level using Microsoft Excel. Additionally, three-way trade-offs and synergies among carbon storage, timber volume, and water yield provision levels were then graphed and assessed using R's (Version 3.0.2; <http://www.Rproject.org/>) scatterplot3d package.

3. Results Findings

indicate an overall reduction in total carbon storage and timber volume and an increase in water yield in both the Pensacola and Lower Suwannee river watersheds as a result of urbanization and associated forest cover loss. Indeed, these spatiotemporal changes were different in the two watersheds given their inherent land use-cover compositions during the 2003 to 2060 analysis period. As we detail below, ecosystem drivers affected all three services, except for timber in the Pensacola watershed. Tradeoff interactions among the three ecosystem services were also evident during the analysis period except for a weak interaction between water yield and both carbon storage and timber volume in 2060 in the Pensacola Bay watershed.

3.1. Land use and land cover map in 2003 and 2060

The Pensacola Bay watershed was more urbanized than the Suwannee watershed in 2003. As expected, Pensacola Bay was more impacted by

urbanization in 2060. There was a reduction in forest areas of 21% due to urbanization, whereas the Lower Suwannee River watershed experienced a decrease of 12% of forest areas due to the development of new urban areas.

3.2. Carbon storage in 2003 and urbanized 2060

The total carbon storage from 2003–2060 was reduced by 19% in the Lower Suwannee River (Table 1) and this corresponded with a 12% reduction in forest area in the urbanized 2060 scenario. There was a reduction in mean carbon stores as a result of urbanization across all 63 sub-watersheds. The largest difference was observed in the Waccasassa Slough sub-watershed (Id 49, Fig. 2), where the mean carbon storage decreased from 183 to 156 Mg/ha (15%) as a result of a 14% decrease in forest area and 40% increase in urban areas. Fig. 2 shows the range of changes in high, moderate, and low change categories. Differences in mean carbon storage between all the same sub-watersheds in 2003 and 2060 were statistically significant. In the Pensacola Bay, the total carbon storage was reduced by 26% during 2003–2060 (Table 1) and forest area was reduced by 21% – assuming 5% less carbon – in 2060. The mean carbon storage was 148 Mg/ha in 2003 and 137 Mg/ha in 2060. At the sub-watershed level, the largest difference was observed in the Fundy Bayou Williams Creek Frontal sub-watershed (number 11, Fig. 3), where mean carbon storage dropped from 162 Mg/ha to 144 Mg/ha; forest area was reduced by 34% and urban area increased by 36%. There was a statistically significant reduction of 3–18 Mg/ha in mean carbon storage in all 15 sub-watersheds.

3.3. Timber volume in 2003 and urbanized 2060

Urbanization in the Lower Suwannee watershed produced a decrease of 2,300,000 m³ in the timber net volume in 2060, as a result of the loss of 22,600 ha of forest (Table 1); this despite an increase in timber volume of 1.7% used in the analysis (Prestemon S. Delphin et al. / Land Use Policy 54 (2016) 188–199 193 Fig. 2. Range of change in mean carbon storage (A) and water yield (B) between 2003–2060, by sub-watershed, in the Lower Suwannee River, Florida U.S. 194 S. Delphin et al. / Land Use Policy 54 (2016) 188–199 Fig. 3. Range of change in mean total carbon (A) and mean water yield (B) between 2003 and 2060, by sub-watershed, in the Pensacola Bay watershed, Florida U.S. S. Delphin et al. / Land Use Policy 54 (2016) 188–199 195 Table 1 Total carbon storage (1000 Mg), mean carbon storage (Mg/ha), timber net volume (1000 m³), mean timber net volume (m³/ha), water yield (103 m³) and mean water yield (mm) for the Lower Suwannee River (LS) and Pensacola Bay (PB) watersheds, Florida U.S. Watershed-year Total carbon Mean carbon Timber volume Mean timber volume Water yield Mean water yield LS-2003 36,587 196 20,400 109.5 93,193 1290 LS-2060 30,638 187a 18,100 110.6 96,695 1340a PB-2003 7998 148 3700 68.6 104,631 1500 PB-2060 5867 137a 2900 68.3 108,441 1600a a Statistically significant differences at 95% confidence level. and Abt, 2002a). However, the difference was not statistically significant at a 95% confidence level. Timber volume in the sawlog portion, in 2060 was reduced by 1,940,000 m³ (Table 1). At the sub-watershed level, the largest decrease (17%) in mean timber net volume was for the Waccasassa Slough sub-watershed (number 49, Fig. 2); that had a decrease in forest area of 14% and increase in urban areas of 40%. In Pensacola Bay, there was a reduction of 11,500 ha of forest cover that resulted in a decrease of 800,900 m³ of timber net volume in year 2060 (Table 1). Sawlog timber volume was reduced by 22% in 2060 as a result of the forest area reduction. In 2060 there were no statistically significant decreases in the mean timber net volume at the sub-watershed level. However, 2 of 15 analyzed sub-watersheds did show a non-statistically significant increase. These two sub-watersheds were not significantly affected by the projected urban areas. The largest difference (6 m³/ha) was in the Williams Creek-Oriole Beach Frontal sub-watershed (Id 11, Fig. 3); a decrease of 34% in forest area and an increase of 36% in urban areas.

3.4. Water yield in 2003 and urbanized 2060

Mean annual water yield for 2003 in the Lower Suwannee River watershed was 1290 mm and water yield

volume was 12,900 m³/ha. However, in 2060, the mean simulated precipitation and PET were higher as were volume (13,400 m³/ha) and mean annual water yield (1340 mm); the latter being statistically greater. At the sub-watershed level, there were also increases in water yield, precipitation, and PET in 2060. The largest difference in mean annual water yield during the analysis period were in the Barnett Creek sub-watershed (number 1, Fig. 2) which also exhibited an increase of 152 mm/year and the largest increase in precipitation in 2060; however no discernable changes in LULC were observed. In the 63 sub-watersheds there was statistically significant increased water yield per hectare in 2060. We note that uncertainties in future climate multi-decadal projections are currently large, and thus might mask an important effect of this land use change. There were statistically significant differences in the mean water yield between sub-watersheds in 2003 and 2060. In 2003, the annual water yield in the Pensacola Bay was 1,520 mm and the average water yield per hectare was 15,300 m³. However, in 2060 the assumed model precipitation and PET were higher as were annual water yield at 1,600 mm and water yield per hectare at 15,800 m³. At the sub-watershed level, the largest increase in simulated mean annual water yield was observed in the Bayou Cico-Bayou Garcon Frontal sub-watershed (number 15, Fig. 3) that registered the largest increase in precipitation in 2060 and there was a high positive correlation ($r = 0.969$) between the difference in annual water yield and difference in precipitation. There was also a decrease in the forest area and an increase in the urban areas. We assessed the InVEST model's water yield estimates in the Lower Suwannee River watershed to measure overall performance. Our assessment shows that there were differences between the average water volume at the Gopher gauging station and modeled average water yield. The average value from the model was $5,200,000 \times 10^3$ m³ while the average from the gauging station was $6,700,000 \times 10^3$ m³. However, when values were normalized based on mean and standard deviation, trends between modeled and measured water yield curves were similar (Delphin et al., 2014).

3.5. Three-way interactions between carbon storage, timber volume, and water yield

Although timber volume, carbon storage, and water yield provision levels are provided in Table 1 and Figs. 2 and 3, Fig. 4 shows the magnitude in interactions or trade-offs and synergies among these three services. In the 2060 urbanized scenario, carbon storage and timber volume interactions were lower than in 2003; however, trade-offs with water yield were higher in 2060 (Fig. 4). Overall there was a positive correlation between carbon storage and timber volume interactions (i.e., synergy), and a negative correlation with water yield interactions (i.e., trade-off). Because of forest loss as a direct result of urbanization, it can be surmised that both carbon storage and timber volume will decrease. However, water yield increased when urban areas increased, but the correlation was weakly negative (Table 2).

4. Discussion

4.1. Land use and land cover map in 2003 and 2060

We have used an integrated system of models and geospatial data to demonstrate a more robust approach to projecting the spatio-temporal interactions of ecosystem services in two different subtropical watersheds. We have also identified the areas in both watersheds that were most susceptible to ecosystem service drivers from projected urban development. Results show that integrating available geospatial models and forest inventory and monitoring data can also be used to project the ecosystem service tradeoffs that occur across space and time for different land use policy scenarios. Such knowledge on how land use change alters ecosystem services is fundamental for effective policy making and minimizing the negative effects on human well-being. But, we admit that predicting how future land use change might affect multiple ecosystem services is a difficult and complex task. And thus, no single existing predictive model is likely to be effective at addressing this question. Urbanization of existing forest cover types - not accounting

for urban forest ecosystem service provision - directly affected carbon storage, timber volume, and water yield. Indeed urbanization is the dominant land use change that affects Southern U.S. forests (Alig et al., 2004; Conner and Sheffield, 2006; Nowak et al., 2005; D N Wear and Greis, 2002). Wear and Greis (2002) report that 4-9 million hectares of land are likely to be converted to urban land uses by 2060 in the region alone. Thus, it is likely that urbanization will convert many peri-urban forests by 2060, and result in significant decreases in the provision of carbon storage (Timilsina et al., 2011) and timber volume; but increases in water yield. Since LULC - as opposed to seasonal precipitation - input parameters are very sensitive in the InVEST model, one can surmise that urbanization is 196 S. Delphin et al. / Land Use Policy 54 (2016) 188-199

Fig. 4. Ecosystem service interactions among carbon storage, timber volume, and water yield during 2003 - 2060 for the Lower Suwannee (LS; in black bars) and Pensacola Bay (PS; in grey bars) watersheds, Florida U.S. Table 2 Pearson Correlation coefficients between carbon storage, timber volume, water yield and percent (%) forest change during 2003-2060 at the sub-watershed level for the Lower Suwannee River (LS) and Pensacola Bay (PB) watersheds, Florida U.S. %Forest change Carbon Timber Water LS %Forest change 1 LS Carbon 0.98 1 LS Timber 0.99 0.98 1 LS Water -0.86 - 0.93 -0.87 1 PB %Forest change 1 PB Carbon 0.99 1 PB Timber 0.97 0.97 1 PB Water -0.14 -0.13 -0.20 1 more influential than precipitation in affecting water yield in this analysis (Sánchez-Canales et al., 2012)

4.2. Carbon storage in 2003 and urbanized 2060 Mean carbon storage for the Lower Suwannee and Pensacola Bay watersheds were 196 and 148 Mg/ha, respectively, for the year 2003 and were within the range reported by Heath et al. (2011) for forest areas in the United States (162 Mg C/ha and a range of 150-192 Mg/ha). According to Timilsina et al. (2011) mean carbon stores for Florida forests were 165 Mg of carbon per ha in cold spots (i.e., areas of below average carbon storage values) and 177 Mg of carbon per ha in hot spots (i.e., areas of above average carbon storage values). As stated previously, carbon stored in urban areas was not taken into account. Escobedo et al. (2009) presented a comparison of average tree carbon storage densities by urban forests in Florida that ranged 27.4 Mg per hectare in Pensacola and 30.8 Mg per hectare in Gainesville. Thus, using this study's 2003 average forest carbon storage values, the average carbon storage for Pensacola's urban forests (Escobedo et al., 2009), and adding this value to the mean carbon storage estimated in our study, the Pensacola Bay watershed will store about 175.4 Mg of carbon per ha representing about 16% of carbon stored by the watershed's urban forests. Using carbon storage values that include both urban and natural forests, total carbon storage in the urbanized 2060 scenario in Pensacola Bay watershed will decrease by 26% when compared to the 2003 total carbon storage. Thus, including urban forest carbon stores in our analysis; there would still be an overall decrease in total carbon storage in the Pensacola Bay watershed from 2003 to 2060. In Pensacola Bay, the forest area loss due to urbanization was 21% and findings corroborate other studies that show that land use change is an important driver affecting carbon storage and sequestration in ecosystems (Birdsey et al., 2006; Moore et al., 2002; Reyers et al., 2009), in addition to diseases, pests, hurricanes, and wildfires (McNulty, 2002).

4.3. Timber volume in 2003 and urbanized 2060 The Lower Suwannee had 11% less timber volume in 2060 compared to 2003; however, Pensacola Bay had a larger decrease (21%). As previously mentioned, the Pensacola Bay is more influenced by urbanization. Indeed Conner and Hartsell (2002) report that Florida has lost most of its forest areas due to urbanization. Hodges et al. S. Delphin et al. / Land Use Policy 54 (2016) 188-199 197 (1998) found that the forest area was reduced by urbanization by 25% compared to this study's 12% and 21% in the Lower Suwannee and Pensacola Bay watersheds respectively.

4.4. Water yield in 2003 and urbanized 2060 In both

watersheds, water yield was higher in 2060 as were precipitation and PET. Based on CSIRO Mk3 and other studies, both precipitation and evapotranspiration should increase during this study's analysis period (Alvarez et al., 2001; Florida Oceans and Coastal Council, 2009; Furniss et al., 2010). Increase in water yield is primarily due to increase in precipitation (Dymond et al., 2012; Sun et al., 2005) and changes in LULC. Again, studies have documented lower water yield in forested areas (Bosch and Hewlett, 1982; Dymond et al., 2012; Hibbert, 1965). In many sub-watersheds in our study, the increase in water yield was the result of the increase in precipitation, decrease in potential evapotranspiration, and decrease in forest areas (Dymond et al., 2012). The increase in water yield can however also have negative effects (i.e., ecosystem disservices), such as increased flood risk, nutrient transport, decreased water quality, and sedimentation of water bodies. Differences in observed water yield between the InVEST model estimates and measured gauging station data have been previously observed by Delphin et al. (2014). These differences could be related to groundwater flow since according to Katz et al. (1997) watersheds in the study area are characterized by the presence of karst features that enable ground and surface-water interaction. The fact that the InVEST model does not account for these groundwater processes, should be taken into consideration where model estimates are important for management and policy actions, thus further calibration of the InVEST model or use of other models is warranted (Delphin et al., 2014).

4.5. Three-way interactions between carbon storage, timber volume, and water yield Trade-offs among ecosystem services occur when the increase in the provision of one service can decrease the provision of another service (Bottalico et al., 2016; Cademus et al., 2014; Rodriguez et al., 2006). In general, increases in provisioning services will result in a decrease in regulating services (Dymond et al., 2012; Raudsepp-Hearne et al., 2010). Therefore, as also reported by Cademus et al. (2014), when forests are managed for increased carbon storage, water yield can be a trade-off as observed in our study. Also, if timber production is prioritized, the timber-carbon relationship will be negative; as timber harvesting will result in a decrease in biomass and carbon storage in live trees. Similarly, an increase in water yield due to forest loss can be considered either a win-lose (i.e., trade-off) or win-win (i.e., synergy) situation depending on the socioeconomic value placed by society to specific ecosystem services in a watershed (Kreye et al., 2014). For example if there is a demand for irrigation water, more timber will result in less water yield (win-loss); However, decreased timber from logging results in more water yield that can result in either more water for irrigation or increased runoff, sedimentation, and flooding (Dymond et al., 2012).

4.6. Limitations and future research There are limitations to our modeling approach. First, the 2003 FVLC map has been revised to create a more current statewide Cooperative Land Cover, version 2.3 (Florida Natural Areas Inventory, 2012). But because of the time of release it could not be used for this analysis. Also, the watersheds will likely experience interactions and feedbacks that are not captured when modeling for each ecosystem service. For example, non-linear threshold effects or urban development patterns are difficult to predict and model as are changes in future economic and social conditions at both the local and regional levels (Yue et al., 2013). Additionally, there are major uncertainties about regional precipitation patterns and temperature projections associated with anthropogenic climate change (Giorgi and Francisco, 2000). These direct effects and additional potential climate feedbacks to the vegetation, such as changes in fire risk and primary productivity, add to the uncertainty in predicting future changes in ecosystem service provision. Our approach and findings however indicate several potential areas for future research. First, other direct and indirect drivers such

as wildfires, drought, pest outbreaks, sea level rise, climate change or even increased energy and real estate prices could be modeled individually or coupled to determine the effects on ecosystem service bundles of interest. Second, determining what key ecosystem services - or disservices- and their supply level are in demand (or not) by beneficiaries according to relevant scales and contexts is important. This is particularly useful for determining when drivers will result in trade-offs or synergies and what specific economic valuation methods to apply. Finally, approaches similar to ours could be integrated into applied geospatial tools and modeling scenarios to better visualize the effect of drivers on ecosystem service outcomes. These could also be used for spatially explicit analyses of the role of conservation or land use policies and activities (e.g., vegetation removal, urban development, land use ordinances) on long-term ecosystem service provision.

5. Conclusions

Our findings highlighted the influence of a direct driver, urbanization, on trade-offs and synergies among ecosystem services during key periods of past, existing and projected increased urbanization rates. But, even though our findings show that there were - and will be - trade-offs among ecosystem services, some planning and management alternatives can be used for reducing these negative interactions. Thus, a key challenge when formulating policies for the provision of ecosystem services is identifying, defining and targeting optimal provision levels. As such, understanding the response of ecosystems to different drivers - using enhanced modeling alternatives and data- is key to finding these optimal levels and approaches such as ours can be used in identifying the areas that are the least "win-win" and most "lose-lose" and least susceptible to detrimental urbanization effects. An integrated system of models, including InVEST, regional land cover change and climate model alternatives, coupled with spatial data fusion from soil surveys, forest inventories, land cover classifications, and climate data sets can be used to project changes in ecosystem services associated with urbanization. This integrated approach is not exclusive to forested subtropical watersheds and can thus be used to better understand and identify the effects of drivers on relevant ecosystem structure (e.g., forest cover, soils, density, etc.) and subsequent functions (e.g., biodiversity, water regulation, primary productivity) and services when formulating policies in other geographic contexts as well. The results can be useful for land planners and managers in identifying specific conservation areas and potentially susceptible areas to forest land cover losses caused by urbanization. In doing so, the effects on ecosystem services can be used to assess different land use policies, management practices, and planning alternatives that minimize these effects. We hope that studies such as ours can be used to not only better understand - and identify - the effects of drivers on relevant ecosystem services, but also to visualize the effects of policy decisions using geospatial tools and thus encourage the public and private sectors to participate in land management decisions.

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Appendix A. Carbon pools used in this study

according to US Forest Service Forest Inventory and Analysis (FIA) definitions (Woudenberg et al., 2010). Carbon pools FIA carbon data

Aboveground biomass Carbon aboveground + carbon understory aboveground
Belowground biomass Carbon belowground + carbon understory belowground
Soil organic matter Carbon soil
Dead biomass Carbon litter + carbon standing dead + carbon down dead